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16. ABSTRACT

This literature review summarizes current data on the effects of pesticides and metals on marine organisms, aquatic environmental research methods, bioaccumulation of pollutants by estuarine and marine organisms, and in water-sediment residues and biota. Results of studies of the environmental effects of dredging, dumping, wastewater sludge and complex effluents are reported. The occurrence of diseases and tissue abnormalities in fishes is discussed, as well as research related to marine oil pollution.

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Marine and estuarine pollution

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PESTICIDES

The U. S. Department of Commerce published two annotated bibliographies on the ecology of insecticide water pollution, including marine organisms and ecosystems.[1,2] Jernelov[3] presented a history of chlorinated hydrocarbon pollution in the marine environment. Ernst[4] reviewed the effects of pesticides in the marine environment and observed that estuaries exhibit higher concentrations than open seawater. He concluded that concentrations of a number of pesticides in the open ocean are unlikely to endanger marine life, but that the margin for safety is less in estuaries.

Methods. An electron-capture gas chromatographic method for determining residues of chlordanes in fish and shellfish in the presence of polychlorinated biphenyls (PCBs) and common organochlorine pesticides was described.[5] Mackinnon *et al.*[6] presented a method of preparing reference samples containing defined quantities of pesticides at concentrations below, at, and above those usually found in natural samples. An apparatus designed to produce tidal fluctuations in simulated saltmarsh systems was described by Cripe.[7] The device, used to investigate pesticide dynamics, can produce either one or two tidal cycles per day in laboratory tanks.

Gutierrez-Galindo[8] discussed the use of the mussel *Mytilus californianus* as a bioindicator of water column contamination.[7] Intertidal mussels were transferred to a tautline buoy system, suspended at varying depths, and sampled periodically as part of a monitoring study off the Mexican coast. The use of the mussel *Mytilus californianus*, as an indicator of organochlorine pollution levels in a comprehensive monitoring program of the California marine environment, was described.[9] The importance of defining the problems to be addressed prior to designing monitoring strategies was emphasized.

A quality assurance (QA) program established to evaluate the performance of laboratories investigating the herbicides, atrazine and linuron, in Chesapeake Bay water was reported.[10] Based on the results, suggestions were made for improvements in sample analysis and implementation of future QA programs.

Using the mysid shrimp *Mysidopsis bahia*, Nimmo *et al.*[11] summarized criteria for determining chronic effects of pesticides, compared the sensitivity of mysids with other species exposed to the same chemicals, and discussed acute versus chronic test results with regard to estimating "safe" concentrations of a chemical throughout an entire life cycle. After extensive exposure of various test organisms to a number of pollutants, Adema and Vink[12] reached the following conclusions regarding toxicity:

1. In view of specific sensitivity of certain species to certain substances, the environmental toxicity of substances can be estimated only after a number of tests have been made with test organisms chosen from different animal groups;
2. Mortality, growth, development, or reproductive effects are generally observed at much lower concentrations of test substances in subchronic tests than in acute tests; and
3. The number of different animal species suitable for use in routine reproduction testing is still limited.

An easily manipulable, small, and sensitive test organism suitable for reproduction studies is still needed.

Acute effects. Paraquat, a bipyridyl herbicide, was found to exhibit LT_{50} values of 16, 18, and 10 days for a bony fish, *Mugil cephalus*; a crustacean, *Pagurus* sp., and a mollusk, *Murex brandaris*, respectively, at 1 mg/l.[13] No mortality for any species was observed at 0.1, 0.05, and 0.01 mg/l for up to 3 months' exposure for the fish and 1 month for other species. The toxicities of pesticides to the shrimp *Crangon*

septemspinosa were 10 to 80 times less in sediment than water; it was concluded that compounds absorbed to sediment contributed little towards toxicity.[14] Eversole[15] found that a 96-hour LC_{50} value for the prawn *Macrobrachium rosenbergii*, exposed to Mirex for 24 hours, was 104 µg/l. Long- and short-term effect of pentachlorophenol (PCP), dieldrin, and DCA on the polychaete *Ophryotrocha diadema* were compared by Hooftman and Vink,[16] who concluded that mortality provided only a limited picture of the toxic action of the substances tested. Exposure of long duration reduced the LC_{50} values only slightly. Additional information on acute effects of pesticides is presented in Table 1.[17-20]

Table 1—Acute toxicological effects of pesticides.

Compound	Group	Species	Type[a]	Effective concentration (µg/l)	Temp (°C)	Salinity (ppt)	Reference
Chlordane	Shrimp	*Crangon septemspinosa*	96h LC_{50}-S	2.0	20	—	14
Cypermethrin	Fish	*Salmo salar*	96h LC_{50}-S	2.0	10	—	17
	Decapod	*Homarus americanus*	96h LC_{50}-S	0.04	10	—	17
	Shrimp	*Crangon septemspinosa*	96h LC_{50}-S	0.01	10	—	17
DDT	Shrimp	*Crangon septemspinosa*	96h LC_{50}-S	0.4	20	—	14
Dichloroaniline	Algae	*Phaeodactylum tricornutum*	96h EC_{50}-S, FT	450	—	—	12
	Mollusk	*Mytilus edulis*	96h LC_{50}-S, FT	9 500	—	—	12
	Polychaete	*Ophryotrocha diadema*	7d LC_{50}-S, FT	10 500	—	—	12
	Crustacean	Various	96h LC_{50}-S, FT	2 300–14 000	—	—	12
	Fish	Various	96h LC_{50}-S, FT	2 400–4 600	—	—	12
Dichloroaniline	Polychaete	*Ophryotrocha diadema*	96h LC_{50}-S	400	21	33	16
Dieldrin	Shrimp	*Crangon septemspinosa*	96h LC_{50}-S	0.4	20	—	14
Dieldrin	Algae	Various	96h EC_{50}-S, FT	>100	—	—	12
	Mollusk	*Mytilus edulis*	14d L_{50}-S, FT	>200	—	—	12
	Polychaete	*Ophryotrocha diadema*	14d LC_{50}-S, FT	>100	—	—	12
	Crustacean	Various	48h LC_{50}-S, FT	12–55	—	—	12
	Fish	Various	48h LC_{50}-, FT	4.0–6.5	—	—	12
Dieldrin	Polychaete	*Ophryotrocha diadema*	96h LC_{50}-S	>100	21	33	16
Endosulfan	Shrimp	*Crangon septemspinosa*	96h LC_{50}-S	0.2	20	—	14
Endrin	Shrimp	*Crangon septemspinosa*	96h LC_{50}-S	0.6	20	—	14
Fenvalerate	Fish	*Salmo salar*	96h LC_{50}-S	1.2	10	—	17
	Decapod	*Homarus americanus*	96h LC_{50}-S	0.14	10	—	17
	Shrimp	*Crangon septemspinosa*	96h LC_{50}-S	0.04	10	—	17
Hexachloran	Mussel	*Mytilus galloprovincialis*	96h LC_{50}	4 700–5 511	17	18	18
Pentachlorophenol	Algae	Various	96h EC_{50}-S, FT	200–5 500	—	—	12
	Mollusk	*Mytilus edulis*	96h LC_{50}-S, FT	1 800	—	—	12
	Polychaete	*Ophyrotrocha diadema*	96h LC_{50}-S, FT	1 200	—	—	12
	Crustacean	Various	96h LC_{50}-S, FT	170–16 000	—	—	12
	Fish	Various	96h LC_{50}-S, FT	60–1 600	—	—	12
	Polychaete	*Ophryotrocha diadema*	96h LC_{50}-S	600	21	33	16
Permethrin	Fish	*Salmo salar*	96h LC_{50}-S	12	10	—	17
	Decapod	*Homarus americanus*	96h LC_{50}-S	0.73	10	—	17
	Shrimp	*Crangon septemspinosa*	96h LC_{50}-S	0.13	10	—	17
Slimicide C-30	Crustacean	*Leptomysis mediterranea*	96h LC_{50}-FT	110	14	37.5	19
Sodium Pentachlorophenate	Copepod	*Pseudodiaptomus coronatus*	96h LC_{50}-S	68	20	18	20
Trichloroethane	Algae	Various	96h EC_{50}-S, FT	60 000–260 000	—	—	12
	Mollusk	*Mytilus edulis*	96h LC_{50}-S, FT	110 00	—	—	12
	Polychaete	*Ophyrotrocha diadema*	96h LC_{50}-S, FT	190 000	—	—	12
	Fish	*Pleuronectes platessa* (egg)	48h LC_{50}-S, FT	125 000	—	—	12

[a] FT = Flow through; S = Static.

Sublethal effects. The intrinsic rate of population increase for the estuarine copepod *Eurytemora affinis* was reduced by 5 ppb dieldrin, which was 20% of the LC_{50} value.[21] This decline was attributed to increased time to reproductive maturity, increased time between broods, and increased mortality of pre- and postreproductive females.

Exposure of the marine pulmonate *Onchidium verruculatum* to sublethal concentrations of DDT for 4 to 96 hours significantly increased O_2 consumption.[22] A 24-hour pretreatment with either a monamine depletor or two monamine antagonists produced declines in O_2 uptake following DDT exposure.

Kuwabara *et al.*[23] found that 10 mg/l EPN inhibited the hatchability of *Artemia salina.* However, parathion, diazion, malathion, DDT, dieldrin, BHC, and captan had no effect on hatching. Size- and sex-dependent adverse effects of atrazine on the fiddler crab *Uca pugnax*, at concentrations as low as 100 mg/l, were observed.[24] The toxicity of atrazine to fiddler crabs, however, was strongly seasonally dependent. It was concluded that exposure to atrazine, even if present at concentrations over 1 000 times greater than expected in runoff from agricultural land, would have no significant direct effect on the size classes tested. Lucu *et al.*[19] determined that the biocide-slimicide C-30 blocked embryonic development of the mussel *Mytilus galloprovincialis* at low concentrations. The effect was evident at 0.02 mg/l, a concentration 1 000 times less than recommended by the manufacturers as the concentration necessary to prevent fouling in industrial applications.

Forney[25] found 50% growth inhibition at atrazine concentrations of 13 ppb for elodea, 163 ppb for wild celery, 474 ppb for redhead grass, and 1 104 ppb for Eurasian watermilfoil. When low atrazine concentrations were present in both water and soil, the water concentration determined the toxic effect. The author concluded that atrazine concentrations detected in Chesapeake Bay were not likely to have caused the observed decline of submerged plants in the bay. Plumley and Davis[26] reached the same conclusion from their studies on salt marsh algae, finding that 2.2 mg/l reduced the rate of photosynthesis, chlorophyll content, and cell numbers in cultures of *Nitzschia sigma* and *Thalassiosira fluviatilis.* Diatom species-diversity was not affected at the same concentration. Using a tenfold safety factor below the least-effect level, it was concluded that a maximum safe level for atrazine in the saltmarsh would be 10 ppb.

Tagatz and Ivey[27] found that benthic community structure was altered significantly by 0.1 μg/l fenvalerate in field studies and 1.0 μg/l in laboratory communities. The abundance of amphipods was significantly altered in both situations, whereas both mollusks and annelids were not affected at 1.0 μg/l in the laboratory and 10.1 μg/l in field. Pentachlorophenol significantly altered the structure of estuarine communities at 141 μg/l (field) and 140 μg/l (laboratory).[28] A 96-hour exposure to 0.31 to 0.56 μg/l methyl parathion and 0.078 to 0.18 μg/l phorate reduced the swimming stamina of the crustacean *Mysidopsis bahia.*[29] Because these results were similar to those obtained in a 28-day exposure, they suggested that this test has potential for use as a rapid screening bioassay.

Microbiology. Jones and Hood[30] determined the effects of seven thiophosphorus pesticides and their degradation products on estuarine ammonium oxidizers and concluded that parent compounds had little effect, but certain decomposition products may have reduced ammonium oxidation significantly, especially in sediments under low O_2 levels.

Kepone at concentrations below 0.01 mg/l did not significantly affect heterotrophic activity in Chesapeake Bay water.[31] Above 0.2 mg/l, however, amino acid uptake and direct viable cell counts were inhibited. The effects of carbaryl, methyl parathion, and pentachlorophenol on microfungal succession in estuarine microcosms were determined by Cook *et al.*[32] Resident fungi was succeeded by *Fusarium* in the carbaryl-treated microcosm, by a fungus in the *Penicillium chrysogenum* series in the methyl-parathion-treated microcosm, and by a fungus in the *P. canescens* series in the PCP-treated microcosms.

Fate. Christiansen and Costlow[33] found that it took 8 weeks to degrade a 10-ppb solution of Dimilin to a concentration that did not affect survival of crab larvae of *Rhithropanopeus harrisii.* When they related this relatively slow degradation rate to toxicity data, the authors concluded that extreme caution is indicated in using Dimilin in estuarine areas where crab larvae occur.

The persistence of parathion, methyl parathion, and fenitrothion in seawater was affected mostly by water quality and light, and partially by temperature, but not by suspended solids and vaporization. Microbial degradation of phosphorus insecticides in seawater was largely suppressed by high salt concentration.[34] Weber and Rosenberg[35] found no significant degradation of toxaphene over 21 days in estuarine sediment and when the sediment was exposed to the action of microorganisms for 1 or 2 days; its toxicity to *Artemia salina* did not change significantly. Eder[36] found that 2,4,5-trichlorophenoxyacetic acid (2,4,5-T) underwent bacterial degradation in anaerobic estuarine sediment.

Field studies over a 2-year period by Orndorff[37] indicated that the number and types of Kepone-resistant bacteria present in a given environment reflected the degree of anthropogenic contamination as well as the major indigenous, gram-negative genera of the sampling sites. He concluded that soluble concentrations of Kepone currently detectable in marine waters would have minimal effects on heterotrophic uptake and cell viability.

Bioaccumulation. The alga *Prototheca zopfii*, exposed to 1 mg/l Kepone for 2 hours, bioaccumulated but did not biodegrade the pesticide, and removed it from solution with an efficiency similar to that of activated charcoal.[38] Roberts and Leggett[39] reported that blue crab *Callinectes sapidus* from the lower James River accumulated more Kepone than crabs from the lower Chesapeake Bay. Extruded eggs, a major route of Kepone clearance from female blue crabs, contained approximately three times more Kepone than backfin muscle.

Duncan *et al.*[40] found a direct relationship in the Los Angeles Harbor between concentrations of DDT and organism trophic level; that is, the higher the trophic level, the greater the total DDT concentration. For example, DDT residues in white croaker and halibut were 5 to 20 times above concentrations found in the clam and anchovy. DDT bioaccumulation from water by the green cod was greatest in liver tissue, followed by gills and muscles. Liver DDT depurated when the fish was placed in clean water.[41]

Harding *et al.*[42] determined DDT was accumulated by the marine copepod, *Calanus finmarchicus*, when fed contaminated phytoplankton, *Thalassiosira weisflogii.* At a phyto-

Table 2—Bioconcentrations factors (BCF) for selected pesticides.

Pesticide	Organism	Source	BCF	Exposure time	Reference
Endrin	Lobster (*Homarus americanus*)	Food	0.41	14 days	43
Mirex	Salmon	Water	84	96 hours	44
Mirex	Salmon	Food	0.135	28 days	44
Pendimethalin	Sea nettle (*Chrysaora quinquecirrha*)	Water	0.42	16 hours	45

plankton density of 60 μg C/l, the copepod retained 60 to 70% of the DDT ingested. At 600 μg C/l, as little as 10% was retained. Additional bioaccumulation information is presented in Table 2.[43-45]

Residues. Dissolved BHC in the Po and Adige Rivers of Italy in 1977 to 1978 ranged from 1 to 37 ng/l and was present in all samples.[46] The average concentration of BHC in the Po River increased from 3.5 ng/l in 1969 to 98.0 ng/l in 1971 to 1973, then decreased to 10.2 ng/l in 1977 to 1978. This increase corresponded to sales, which doubled from 1969 to 1974, then ceased owing to a usage ban.

Atrazine residues were detected year-round in the Rhode River Estuary on the Chesapeake Bay and in rainwater. Water residues ranged from 6 to 190 ng/l; rainwater residues, from 3 to 2 190 ng/l. Although high atrazine concentrations were detected in winter rainfall, these did not result in similarly higher atrazine concentrations in estuarine receiving waters. The author concluded that atmospheric atrazine input was not the dominant source for this rural estuary, which receives atrazine in runoff.[47]

Lunsford[48] found that Kepone in James River estuary water ranged from 0 to 1.2 ppb in 1976 to 78. Surface and bottom-water concentrations were similar, and residues in the water column and underlying bed sediment were significantly correlated. DDE in sediment from San Pedro Basin in the Southern California Bight measured up to 2 400 ng/g in the top 50-mm core section. The 45- to 50-mm core segment was deposited in the 1945 to 1950 period when DDT came into common use. In contrast, DDE occurred only in the San Nicolas Basin sediment, because this basin is a greater distance from anthropogenic inputs.[49]

Brunn *et al.*[50] found that in no cases were HCB, HCH, and Lindane concentrations in Mediterranean fish, shellfish, crustaceans, and mollusks above the maximum permitted concentrations. DDT and HCH were detected in the surface waters of the North Pacific and Indian Oceans at concentrations of 0.01 to 1.0 ng/l and 1.0 to 10 ng/l, respectively.[51] Samples of sediment and bottom-dwelling fishes, crabs, shrimp, clams, and worms were collected at quarterly intervals from urban embayments in Puget Sound. Community characteristics of the sediment-associated invertebrates and bottom fish were defined and their tissue analyzed for pesticides and other chemicals. Although major differences existed between concentrations of contaminants in sediment and biota from non-urban and urban areas, embayments removed from industrial influence were not free of contaminants.[52]

Duinker *et al.*[53] measured dissolved and suspended organochlorine concentrations in the freshwater and estuarine portions of the River Varde A of Denmark. A number of compounds were detected in the river but only hexachlorocyclohexane in the estuary. The amount of each isomer in solution was larger than in suspension in all samples and no evidence was found for exchange of these compounds between solution and suspension during estuarine mixing. Additional information on biotic pesticide residues is provided in Table 3.[54-67]

MICROBIOLOGY

A workshop on the microbiology of polluted waters with reference to scuba diving operations was cosponsored by the National Oceanic and Atmospheric Administration and the National Institute for Occupational Safety and Health. Proceedings of the workshop were published in *Mar. Technol. Soc. J.* (volume 15, issue 2).

The occurrence and distribution of halophilic vibrios was determined in a variety of brackish and marine waters.[69,70] Several studies demonstrated the presence of *Vibrio cholera* in shellfish, sediments, and waters of estuaries on the east coast of the U. S.[71,72] None of the strains found in freshly harvested oysters from one study were enterotogenic, however.[73] It was proposed that *V. cholera* survives in temperate estuaries because it has been recovered from areas with no prior evidence of wastewater contamination or outbreak of cholera infection. *Vibrio vulnificus*, another human pathogen found in shellfish, was inactivated in contaminated oysters that were stored on ice.[74]

Numerous reports on vibrio diseases in fish and shellfish provided further indication of the need to understand the ecology of this group of bacteria in coastal waters.[75-77] A newly described species, *Vibrio damsela*, was shown to cause lesions in rocky reef fishes in southern California.[78] A *Vibrio* species different from *V. anguillarum* and *V. parahaemolyticus* was found to cause disease in cultured juvenile lobster.[79] An actinomycete was also described in connection with endemic diseases of cultured shellfish around Long Island, N. Y.[80]

Obligate anaerobes were surveyed in polluted waters.[81] A small fraction (2.4%) of sediments and seafoods sampled in fishing areas of western Indonesian waters were positive for *Clostridium botulinum*.[82] Other coastal-water and seafood microorganisms that are a public health concern included *Aeromonas hydrophila*[83] and certain yeasts.[84]

The distribution of "indicator" bacteria was reported in waters along the Sicilian coast[85] and in waters and sediments

Table 3—Pesticide residues: biota.

Compound	Species	Location	Residues	References
Aldrin	Mussel (*Mytilus galloprovincialis*)	North Aegean Sea	1 ppb	54
	Crab (*Carcinus mediterraneus*)	North Adriatic	16 ng/g	55
	Holothurian (*Holothuria* sp.)	North Adriatic	12 ng/g	55
DDE	Oysters	New Zealand Coast	0.003–0.005 μg/g	56
DDT	Herring (*Clupea harengus*)	Finland Coast	0.07 μg/g	57
	Common dolphin (*Delphinus delphis*)	California Coast	13–36 mg/l	58
	Striped dolphin (*Stenella coeruleoalba*)	Eastern Pacific	2.6–40 mg/l	58
	Mussel (*Mytilus edulis*)	Scottish Coastal Waters	18–123 ng/g	59
	Harbour Seal (*Phoca vitulina*)	Dutch Wadden Sea	12–47 mg/kg	60
	White-beaked dolphin (*Lagenorhynchus albirostris*)	North Sea	34–71 mg/kg	60
	Mussel (*Mytilus galloprovincialis*)	North Aegean Sea	23 ppb	54
	Mussel (*Mytilus viridis*)	Thailand Estuaries	0.032–0.042 mg/kg	54
	Euphausiids (*Euphausia* sp.)	Mediterranean Sea	n.d.–73 μg/kg	62
	Polychaete (*Alciopa cantrainii*)	Mediterranean Sea	35 μg/kg	62
	Decapod (various)	Mediterranean Sea	4.5–35 μg/kg	62
	Fish (*Myctophum glaciale*)	Mediterranean Sea	11–69 μg/kg	62
	Mussel (*Mytilus galloprovincialis*)	North Adriatic	10 ng/g	55
	Crab (*Carcinus mediterraneus*)	North Adriatic	20 ng/g	55
	Fish (various)	England/Wales Coast	0.001–1.8 mg/kg	63
	Shellfish (various)	England/Wales Coast	0.001–0.04 mg/kg	63
	Harbour Seal	Wadden Sea	8.5–47.3 μg/g	64
	Euphausids	Northeast Pacific Ocean	0.030 mg/l	65
	Fish (various)	Northeast Pacific Ocean	n.d.–76.3 mg/l	65
	Gag (*Mycteroperca microlepis*)	Northwest Atlantic and North Gulf of Mexico	0.017–0.050 mg/l	66
	Black grouper (*Mycteroperca bonaci*)	Northwest Atlantic and North Gulf of Mexico	0.003–0.020 mg/l	66
	Red grouper (*Epinephelus morio*)	Northwest Atlantic and North Gulf of Mexico	n.d.–0.025 mg/l	66
	Red snapper (*Lutjanus campechanus*)	Northwest Atlantic and North Gulf of Mexico	n.d.–0.322 mg/l	66
	King Mackerel (*Scomberomorus cavalla*)	Northwest Atlantic and North Gulf of Mexico	0.009–0.996 mg/l	66
	Spanish mackerel (*Scomberomorus maculatus*)	Northwest Atlantic and North Gulf of Mexico	0.008–0.319 mg/l	66
Dieldrin	Fish (various)	England/Wales Coast	0.001–0.21 mg/kg	63
	Shellfish (various)	England/Wales Coast	0.002–0.59 mg/kg	63
	King Mackerel (*Scomberomorus cavalla*)	Northwest Atlantic and North Gulf of Mexico	n.d.–0.026	64
	Spanish Mackerel (*Scomberomorus maculatus*)	Northwest Atlantic and Gulf of Mexico	n.d.–0.014	64
	Mussel (*Mytilus edulis*)	Scottish Coastal Waters	6–178 ng/g	59

Table 3—(Continued)

Compound	Species	Location	Residues	References
	Harbour Seal (*Phoca vitulina*)	Dutch Wadden Sea	0.1–0.71 mg/kg	60
	White-beaked Dolphin (*Lagenorhynchus albirostris*)	North Sea	5.5–12.4 mg/kg	60
Endrin	Red Snapper (*Lutjanus campechanus*)	Northwest Atlantic and North Gulf of Mexico	n.d.–0.003 mg/l	66
	King Mackerel (*Scomberomorus cavalla*)	Northwest Atlantic and North Gulf of Mexico	n.d.–0.014 mg/l	66
	Spanish Mackerel (*Scomberomorus maculatus*)	Northwest Atlantic and Gulf of Mexico	n.d.–0.026 mg/l	66
Heptachlorepoxide	Harbour Seal (*Phoca vitulina*)	Dutch Wadden Sea	0.68–1.7 mg/kg	60
	White-beaked Dolphin (*Lagenorhynchus albirostris*)	North Sea	0.71–1.3 mg/kg	60
Hexachlorobenzene	Mussel (*Mytilus edulis*)	Scottish Coastal Waters	6–19 ng/g	59
	Mussel (*Mytilus galloprovincialis*)	North Aegean Sea	0.9 ppb	54
	Oyster (*Crassostrea virginica*)	Galveston Bay	0.31–1.37 ng/g	67
Kepone	Blue Crab (*Callinectes sapidus*)	James River/Chesapeake Bay	0.01–0.55 μg/g (eggs)	68
PCP	Oyster (*Crassostrea virginica*)	Galveston Bay	3.40–8.30 ng/g	67

of the North Sea near Ostend.[86] Improvements on the recovery and detection of indicator bacteria and viruses from water and shellfish continued to appear in the literature.[87–93] Caution was advised when comparing identification of coliform bacilli based on different diagnostic keys. Laboratory studies demonstrated the inhibitory effect of sunlight on *Escherichia coli* in seawater.[94,95]

Methods were compared for efficiency in recovering nematode parasites from flatfish.[96] Fish from San Francisco Bay contained higher concentrations of nematodes than those delivered to markets in the Boston area.

Babich and Stotzky[97] reviewed the environmental factors that influence toxicity of heavy metals in microorganisms. A list of publications on the interactions between microorganisms and metals is given in Table 4. A large number of tin-resistant bacteria were recovered from water and sediments in Chesapeake Bay.[105] Factors other than tin concentration seemed to control tin tolerance in these bacteria. Methylation of organotin compounds in estuarine sediments occurred through both biological and abiological processes.[106]

The speciation of metals was also shown to be influenced by bacterial metabolites. Hydrogen sulfide diffusing from polluted harbor sediments caused significant reduction of Cr(6) to Cr(3) in overlying waters.[100] Attachment of Mn-oxidizing bacteria to surfaces enhanced their capacity to precipitate the metal from solution.[102] Several studies demonstrated the role of bacteria in the transfer of metals into marine food webs.[99,100,104] A microplate technique was described that enabled simple and rapid determination of metal accumulation by algae.[107]

The effects of petroleum hydrocarbons on various microbial

Table 4—Microbial interactions with heavy metals.

Element	Habitat/conditions	Reference
Cadmium	artificial	98
Chromium	crab (*Helice crassa*)	99
	Otago Harbor, N.Z.	100
Iron	*Spartina alterniflora* detritus	101
Manganese	*Spartina alterniflora* detritus	101
	artificial surfaces	102
Mercury	Dormant/active marine bacterial populations food web, protozoa, copepods	103 104
Tin	Chesapeake Bay water and sediments	105, 106
Zinc	*Spartina alterniflora* detritus	101

processes were determined in Alaskan coastal waters. Griffiths *et al.*[108,109] reported the acute and long-term effects of crude oil on glucose and glutamate uptake by natural microbial populations in the water and sediments of the Beaufort Sea and Cook Inlet. Long-term exposure of shelf sediments to crude oil also reduced nitrogen fixation and denitrification activities of the indigenous microbial populations.[110]

The characteristics of hydrocarbon degradation by microbial populations were reviewed by Atlas.[111] Button *et al.*[112] described a method for measuring kinetics of dissolved nonpolar hydrocarbon metabolism by microbial populations in seawater. The technique demonstrated toluene oxidation rates of 1 to 30 pg/l · h at 0.5 μg/l toluene in the Port of Valdez.[113] Glass-capillary gas chromatography was used to determine microbial degradation of the aromatic and saturated fractions of Prudhoe Bay crude oil in waters along the Washington coast.[114] Nitrogen and phosphorus supplementation enhanced the degradation of these oil fractions. The oxidation of naphthalene by cyanobacteria and microalgae was also described.[115]

The presence of certain cellular components were related to hydrocarbon degradation. The presence of unsaturated mycolic acid in cells of a bacterium isolated from Arctic seawater facilitated utilization of diesel fuel at low temperatures.[116] The incidence of plasmids in marine *Vibrio* spp. isolated from an oil field in the northwestern Gulf of Mexico did not correlate with the ability of cells to oxidize hydrocarbons or tolerate high concentrations of heavy metals.[117]

The effects of various chemical pollutants on the production of marine bacterial metabolites were reviewed by Aubert *et al.*[118] The bioconcentration of organic pollutants by microorganisms in aquatic environments was also reviewed.[119] Alexander[120] described the types of biochemical transformations carried out by aquatic microorganisms on chemicals of environmental concern but pointed out that there are some new synthetic organic compounds that are degraded very slowly or not at all. In one study, p-aminobenzoate breakdown by bacteria was accelerated in the presence of protozoa.[121] A turnover time of approximately 1 year was estimated for biphenyls and chlorobiphenyls in seawater from determinations based on microbial degradation rates.[122] Taylor *et al.*[123] described the potential for the degradation of plasticizers such as phthalic acid esters by bacteria in sediments and waters of the Mississippi River Estuary.

The effects of complex mixtures of pollutants on microorganisms were determined in several studies. The application of wastewater sludge to the soil of a *Spartina* marsh inhibited denitrification.[124] A study of the bacterial population structure revealed increased numbers of culturable bacteria as well as a greater contribution of gram-positive bacteria in waters around a pharmaceutical waste dump site in the Atlantic Ocean than in surrounding waters.[125] In another study, microorganisms were used to monitor the efficiency of sand capping dredge spoils from New York Harbor.[126] Actinomycetes were suggested as possible indicators of upward carriage of material through the cap.

DISEASES AND ABNORMALITIES

Two investigations focused on the structure and function of ovaries in fish from potentially contaminated sites. Stott *et al.*[127] examined ovaries from 11 species of marine fish from control sites and from gas and oil platform areas. Although there was a similar degree of atresia of ova and follicles and a similar extent of parasitic involvement at control and platform sites, frequencies of ovarian acidophilic cells, chromatophores, and leukocytic foci were higher at platform than control sites (68 versus 17%, 57 versus 11%, and 30 versus 0%, respectively). Shackley *et al.*[128] observed a major transport of copper and cadmium from liver to ovaries in a British blenny, *Blennius pholis*, during egg production and, subsequently, a major expulsion of the metals during spawning. The effects of this metal burden on embryos are not known.

In Japan, studies were continued in an attempt to isolate mutagenic substances that might be implicated in a high incidence of skin melanoma in sea trout (*Nibea mitsukurii*) near pulp mill outfalls.[129,130] At the site with the highest melanoma frequency (Shinguu), sediments produced the highest DNA damage in Ames tests and contained at least four identifiable mutagenic compounds (2,4,6-trichlorophenol; 3,4,5,6-tetrachloroguiuacol; flouranthene; and pyrene). In addition, the livers of both normal and diseased fish from Shinguu, and diseased fish from other sites, contained DNA-damaging material. Further, a DNA-damaging chemical, 9,10-epoxystearate, was isolated from a subfraction of the liver ether extracts of diseased fish, but not from normal fish. In contrast to the organic contaminants, metals in sediments showed an inverse relation with disease prevalence and mutagenicity.[129] The relation between fish kills and toxic dinoflagellate blooms was discussed by White,[131] who demonstrated a direct transfer of *Gonyaulax* toxin to feeding zooplankters and residual storage of the toxin following bloom termination. The fat-soluble agent involved in nonparalytic diarrhetic shellfish poisoning, common in many coastal areas, was found to be produced by the flagellate *Dinophysis* and transmitted to the shellfish; the author called for careful examination of "marginal" episodes of shellfish poisoning and their ecological associations.[132]

DRILLING FLUIDS AND MUDS

Several laboratory studies indicated that marine animals can be affected by exposure to drilling muds above concentrations of 10- to 100-ppm solids (dry wt).[133-135] At both 10 and 100 mg/l of a drilling mud, lobsters (*Homarus americanus*) altered their response to food odors.[133] Another mud at 10 ppm did not affect development of fish (*Fundulus*) or sand dollar (*Echinarachinus*) embryos, but effects were noted at 1 and 10 ppt;[134,135] Crawford and Gates[135] speculated zinc may have been the toxic agent. At an operational platform off New Jersey, disposal of 2 160 tons of drilling fluid apparently caused a depression on polychaete abundance, but caused no change in the relative abundance of predatory, deposit-feeding, or suspension-feeding polychaete guilds, despite measurable increases in sedimentary content of clay, illite, chorite, and kaolinite.[136]

DREDGE AND SOLIDS DISPOSAL

The ability of fish and benthic macrofauna to avoid and recover from disposal of dredge material and solid wastes was examined in the U. S.[137,138] Great Britain,[139] and Canada.[140,141]

On the U. S. east coast, the mollusks *Mercenaria, Nucula,* and *Ilyanassa* are capable of recolonizing surface sediments after burial by dredge material.[136] McFarland and Peddichord[138] found that of 16 species of San Francisco Bay fishes and invertebrates, those from muddy bottom habitats were least sensitive, although some open water fish, fouling organisms, and sandy bottom epifauna were most sensitive to suspended clay (4.5 μm Hydrate flat D Kaolin); 100- and 200-hour LC_{50} concentrations range from 6 mg/l for a very sensitive fish (*Cymatogaster*) to 96 mg/l for mussels.[138] Probert[139] followed a decline in benthic diversity and increase in synaptid sea cucumbers for 2 years at a station in a benthic area impacted by decreasing emissions of solids (from 700 000 to 350 000 tons/yr) from one of Britain's major china clay factories; the author stated that suspended solids over the benthic environment were within natural variation and that, instead, major effects were from persistence of sediment instability. However, in a Canadian study, young herring (*Clupea huerengus*) were found to learn to avoid similarly low suspended solids concentrations (9 to 16 mg/l) and that removal of a "dissolved" leachate in dredged material served to lower herring avoidance thresholds.[140] Another study focused on possible effects that would increase deposition of fine material; based on field experiments, increased sedimentation may be sufficient to depress *Corophium* density tenfold and decrease the abundance of *Mya*, thus affecting fisheries and shore bird feeding.[141] Stern and Grant[142] reported that resuspended sediments are capable of extracting Cu, Pb, Zn, Cd, Ni, and Co from 100-ppm aqueous solutions; absorption was strong for Cu, Pb, and Cd; moderate for Zn; and weak for Ni and Co. Absorption also was enhanced by high pH, warm water, and—for Co and Zn—darkness. Based on these observations, the authors suggested that Ni and Co should be of more concern than the other metals in terms of toxic effects on aquatic organisms. At a dredge dumpsite off Honolulu, Albrecht *et al.*[143] found that dumping did not affect the distribution of Cd, Cu, and Zn in control and impacted shrimp (*Heterocarpus ensifer*), but may have been a factor contributing to a possible substitution of Cd for Cu in shrimp at the dredge site.

SURVEYS

Mearns[144] reviewed results of comprehensive ecological studies conducted for the past 10 years near five major wastewater outfalls in the Southern California Bight. Observed ecological effects have included changes in benthic community structure and in the relative abundance of demersal fish near four outfalls; chronic fin erosion at two sites; and contamination of demersal fish and invertebrates by toxic substances at several shelf and intertidal areas. There has been measurable improvement in biological conditions near the outfalls during the period of 1970 to 1980. Twenty-eight Baltic Sea sites from 1920 and 1923 were resampled using similar methods in 1976 to 1977 by Cederwall and Elmgren.[145] Macrofaunal biomass above the halocline (<70 m) increased significantly (median ratio of 4.3) in the latter survey, whereas all sites below the halocline displayed biomass reductions. The authors attributed the biomass changes to eutrophication, resulting in higher food supplies above the halocline and oxygen depletion of deeper waters.

Reish *et al.*[146] described changes in the benthic communities of Los Angeles-Long Beach Harbors over a 28-year period during which a pollution abatement program substantially reduced industrial and domestic discharges. Control of industrial waste discharges during 1968 to 1970 resulted in elimination of very polluted zones in the inner harbor. Subsequent upgrading of wastewater discharges to secondary treatment and diversion of fish cannery wastes to the treatment plant resulted in population decreases of benthos, fish, and birds in the outer harbor. Such decreases were attributed to reduced nutrient input. A new long-term program of monitoring marine resources along the Atlantic coast between Cape Hatteras and the U. S.-Canadian border was described.[147]

Dauer and Conner[148] studied estuarine polychaete assemblages in the vicinity of a secondary wastewater treatment plant outfall in Old Tampa Bay, Fla. At a site about 500 m from the outfall, the polychaete fauna displayed significant enrichment as evidenced by increased biomass, density, and average species number. Species showing the greatest biostimulatory effect were those with benthic larval development or brooding behavior. Elevated chlorophyll *a* concentrations reaching approximately 20 μg/l were recorded in Victoria Harbour, which receives untreated wastewater from Hong Kong.[149] Little evidence of overall changes in taxonomic composition or diversity was noted, except for a summer dominance of *Chaetoceros* spp. in polluted areas.

The zonation and relative abundance of intertidal organisms near a wastewater outfall on the Loughor Estuary, U. K., showed no detectable changes following approximately 1 month of primary effluent chlorination.[150] During the chlorination, there was an accumulation of halogenated C_1 and C_2 hydrocarbons in *Mytilus edulis, Littorina littorea,* and *Fucus vesiculosus.* Within 5 months after cessation of chlorination, the tissue concentrations of halogenated compounds decreased to prechlorination levels.

The density and diversity of intertidal benthic macrofauna were severely reduced, and meiofaunal biomass was enhanced, near an organic waste discharge in the Ems Dollard Estuary (Neth.).[51] Outfall effects were difficult to detect and were relatively confined because of the seasonal nature of waste discharges and the high degree of natural environmental stress in the estuary. A pollution-caused gradient in zooplankton assemblages was documented near the Marseilles-Cortiou wastewater outfall.[152] An area beyond the immediate outfall vicinity to about 500 m was characterized by euryoecious species such as *Acartia clausi, Oithona nana, Euterpina acutiforns,* and *Oikopleuridae.* Although initial reports indicated substantial mortalities of local biota in south San Francisco Bay following a large spill of partially treated wastewater, a field survey of fish, invertebrates, and harbor seals at a nearby slough revealed no obvious short-term effects on abundances.[153]

A survey of benthic macroinvertebrate communities in Oslofjord (Nor.) by Mirza and Gray[154] revealed declining biological conditions owing to organic enrichment and naturally poor flushing characteristics. Use of diversity indices, the log-normal distribution of individuals among species, and multivariate statistical techniques showed a well-defined gradient from polluted conditions in the inner fjord (dominated by *Capitella capitata* and *Polydora* spp.) to normal, highly di-

verse communities near the Drobak sill. The polluted Tees Estuary (U. K.) was found to support a rich infaunal ciliated protozoan assemblage similar to that of unpolluted areas.[155] Lethal toxicity tests did not indicate any increased adaptation to heavy metals by the Tees Estuary strain of *Uronema marinum* (Ciliata). Venec-Peyre[156] reported increased malformations and parasitism in foraminifera collected from the Cale de Dourduff (Fr.). Stress from oil from the Amoco Cadiz spill in an area naturally unfavorable for foraminifera was suggested as the cause. Bourcier[157] documented changes in the macrobenthos of the French Mediterranean coast resulting from harbor construction, wastewater discharges, and changes in wind patterns during the period from 1961 to 1978. Industrial wastes discharged in the Gabes region, Tunisia, have resulted in acidification of nearshore waters and have destroyed the benthic communities over an area of 230 ha.[158]

A study of the distribution of the polychaete *Capitella capitata* in a polyhaline nontidal dock (River Mersey, U. K.) revealed that highest abundances were associated with unstable coarse sediments with low levels of organic matter.[159] Low numbers of *C. capitata* occurred in the organically enriched anoxic silty sands. The data emphasized the need for caution in the use of *C. capitata* as an indicator of organic enrichment, because this opportunistic species is capable of exploiting unstable substrates (caused by pollution or natural factors), but its occurrence in enriched sediments can be limited because of oxygen depletion and sulfide accumulation. Rainer and Fitzhardinge[160] evaluated the effects of environmental variables on macrobenthic community structure in an estuarine system in New South Wales undergoing periodic deoxygenation of its deep waters. Dissolved oxygen (DO) fluctuations were considerably more important than changes in temperature or salinity in controlling species distributions. Diversity (H') and evenness (J') displayed no simple relationships with the degree of environmental stress. Gray[161] discussed the use of the log-normal distribution of individuals among species for detecting pollution-induced perturbations and compared this technique with other related techniques. Lack of fit to the log-normal distribution is probably not a reliable method of detecting pollution effects because heavily stressed communities will return to a good fit after establishing equilibrium. However, initial changes in community structure are indicated by increased skewness, and progressive increases in the standard deviation of the log-normal plot can be used to detect long-term changes.

Raffaelli and Mason[162] suggested the use of the nematode/copepod ratio in meiofaunal communities as an indicator of the degree of organic pollution in sandy intertidal habitats. Samples from heavily polluted beaches in the United Kingdom had ratios exceeding 100, caused by increases in deposit-feeding nematodes and decreases in more sensitive copepods. Warwick[163] indicated that consideration of the metabolic requirements of the different body sizes of copepod and nematode feeding types enables a prediction of copepod/nematode ratios in unpolluted conditions, which can then be compared with measured values. The validity of the nematode/copepod ratio as a generic pollution assessment method was rejected by Coull *et al.*[164] because of among-replicate variability, the potential for nonpollution-related seasonal or tidal changes, and several cited exceptions to nematode dominance at organically enriched sites. In a subsequent response, Raffaelli[165] indicated that nonpollution-induced changes in the nematode/copepod ratio were not responsible for the observed variability, and also calculated ratios for the original sites based on abundances of copepods and epigrowth-feeding nematodes only.

Krieger *et al.*[166] suggested that total body water measurements of mussels may be useful as a noninvasive condition index for pollution monitoring. Evidence was also provided for oxidative metabolism of persistent organic substances in mussels. Widdows *et al.*[167] found that three physiological stress indices (growth efficiency, scope for growth, and oxygen/nitrogen ratio) showed measurable declines in transplanted mussels (*Mytilus edulis*) along a gradient of increasing pollution in Narragansett Bay. A 1-month exposure was sufficient to allow for contaminant accumulation and physiological response of the mussels. Although barnacles displayed identical seasonal breeding patterns in highly polluted, moderately polluted, and relatively clean waters near Bombay, India, Rege *et al.*[168] found that egg production and overall breeding potential were lowest in populations from polluted waters. Nair *et al.*[169] reported that polluted sites near Bombay had higher population fluctuations and lower species diversity of chaetognaths when compared with relatively unpolluted areas. A discharge from a titanium dioxide factory had no effects on the zooplankton near Trivandrum, India.[170]

Sheridan and Badger[171] studied the effects of continuous chlorination on plankton-derived estuarine benthic communities using experimental substrate chambers. Total chlorine-produced oxidant concentrations of 0.47 to 1.41 mg/l resulted in a relatively constant 38 to 44% decline in total individuals with no apparent gradient response. Chlorination had inconsistent effects on colonization and community development, although changes in the abundances of dominant species were detected in pooled data from 12 consecutive studies. Radford and Joint[172] reported on the development and validation of a deterministic ecosystem model, GEMBASE, which has been applied to the Bristol Channel and Severn Estuary, U. K. The model contains 20 state variables and about 150 processes described by first-order differential equations, and was proposed as a research tool and management aid.

RESIDUES AND RELATED TOPICS

Simpson[173] reviewed the effect of Cd in the marine environment. Topics included the distribution, transport, and physical and biochemical effects of Cd on biota. Mollusks concentrated Cd to the greatest extent, especially in the liver and kidneys. Food chain amplification was not evident from whole body analysis. Cd tissue concentrations of 270 μg/g (dry wt) did not affect reproduction in oysters, but placing them in a 0.015 μg/g solution of Cd reduced development of their gametes.[174] Organic substances released by phytoplankton were found to decrease the amounts of Cd and Cu taken up by these plants.[175] Rai *et al.*[176] reviewed the toxicity of metals to algae. Heavy-metal pollution was found to lead to a dominance by a few tolerate species such as *Nodularia*, *Oscillatoria*, *Cladophora*, *Hormidium*, *Fucus*, and *Laminaria*.

In a field study to determine whether or not Cd, Cu, Pb, and Zn accumulated in fish, it was found that the concentration of the metal was related to the invertebrate food item. Fish that fed upon polychaetes had the highest concentration of metals, followed by crustaceans.[177]

The concentration of metals found in marine organisms are presented in Table 5.[178-223] Data on the analyses of marine organisms for the presence of metals and other elements are presented according to the element, group of organisms, and geographical region.

OIL POLLUTION

One hundred and three papers dealing with nearly every aspect of petroleum in the marine environment were presented at the 1981 Oil Spill Conference. Topics covered included dispersants, effluent water studies, cold water studies, numerous case history sessions, platform studies, and contingency planning.[224] Proceedings of an international conference on petroleum in the sea included papers on cost/benefit analysis of environmental management, polar activities, temperate zone, and tropical zone.[225] A bibliography[226] on oil pollution seemed relevant to the Canadian marine environment. Treaties included a monograph on strategies and techniques of oil spill studies[227] and a Gulf of Mexico production field.[228,229] Papers were presented at a conference dealing specifically with consequences of the *Amoco Cadiz* disaster.[230] The significance of personal disposal of spent crankcase oil in coastal towns was discussed.[231]

Residues—analysis. The volume of oil that came ashore during the *Amoco Cadiz* spill was calculated by use of aerial photography; sediment sampling; and accurate measurements of oil depth, and of width and length of oiled beaches.[232] Methanolic-KOH digestion and soxhlet extraction of hydrocarbons from marine sediments were compared.[233] The difficult task of separation and quantification of polynuclear azaheterocyclic hydrocarbons (azaarenes) was facilitated by reversed-phase liquid chromatography.[234] Detection of extremely low levels (parts per trillion) of benzene in seawater by purge-and-trap GC was detailed.[235] Different techniques were compared for the GC analysis of uniform samples of sediments, mussels, fish, shrimp, and green algae.[236] The separation of chromatographic "doublets" of isoprenoid alkanes was shown by Gassman[237] to facilitate the analysis of biogenic vis-à-vis fossil fuel hydrocarbon contamination of seawater samples. Saponification of duck flesh yielded superior reproducibility and extraction efficiency for GC-MS analyses.[238]

Residues—water and sediment. Concentrations of hydrocarbons in Sargasso Sea and Gulf Stream surface waters averaged about 30 kg/l for spring 1978.[239] Pollution by tar balls of Florida beaches [240] and north African shelf surface waters[241] was documented. Two separate works presented detailed characterizations of hydrocarbon vents and formation waters from offshore production operations in the Gulf of Mexico.[242,243] Determinations of volatile hydrocarbon concentrations in surface waters surrounding the *Ixtoc I* well blowout revealed that hydrocarbon losses were principally a result of evaporation that occurred within a few miles downstream of the blowout.[244] Surface microlayer hydrocarbons in Northwest Atlantic shelf waters were enriched relative to bulk water but varied qualitatively, indicating that the use of a "microlayer enrichment factor" for the waters studied may not be appropriate.[245] Pristane, phytane, and C_{16}-C_{32} n-alkanes were among the major components of analyzable dissolved lipids of Gulf of Mexico waters.[246] Particulate lipids (including hydrocarbons) were found to be a complex mixture.[247] Hydrocarbon concentrations, indexed to Ekofisk crude oil equivalents, were reported for waters and sediments of the North Sea, English Channel, Irish Sea, and U. K. estuaries.[248]

Contamination of surficial sediments by natural seepage[249] and refinery effluent[250] hydrocarbons was characterized. Mediterranean Sea nearshore sediments with amounts of total hydrocarbons <20 μg/g (dry wt) were deemed unpolluted.[251] Polycyclic aromatic hydrocarbons were determined in sediments from a Japanese port[252] and two sites in the Baltic Sea.[253]

Residues—fate and weathering. The fate and weathering of petroleum spills in the marine environment were reviewed.[254] A combination of light with water sprayed on an oil surface produced the largest relative decreases in volatile saturate and most aromatic hydrocarbons on Prudhoe Bay crude oil.[255] Laboratory studies showed that volatile hydrocarbon evaporation rates for seawater-in-crude oil emulsion were similar to raw crude.[256] One-fourth the initial amount of radiolabelled benz(a)anthracene introduced into a large-scale model marine microcosm remained unchanged after 230 days.[257] The magnitude of the "salting out" effect of three aromatic hydrocarbons in natural seawater was quantified by Rossi and Thomas.[258]

Residues—biota. Most studies reporting residues of petroleum within marine organisms presented data for several phyletic groups, including hydrocarbons in several species of bottom invertebrates from Antarctica,[259] saturated hydrocarbons in fish and crustaceans of the Mediterranean,[260] and C_{14} to C_{32} hydrocarbons in oysters from the Arabian Gulf.[261] Several species of sea life from San Luis Pass, Tex., were found to lack detectable levels of benzo(a)pyrene.[262] Oil residues in bivalve mollusks and flatfish were correlated with those in associated sediments for the northern Baltic.[263] Concepts of lipid/water equilibration to explain bioconcentration of hydrocarbons in *Mytilus* species were tested *in situ* and seemed to hold for petroleum mixtures commonly encountered in coastal waters.[264] A novel C_{27} stanol was reported by Brassell and Eglinton[265] from deep-sea sediments.

Dispersants. Results and recommendations of a Canadian appraisal of research on oil and dispersants were given by Sprague *et al.*[266] The toxicity of Corexit 9527 for a marine ciliate[267] and pelagic/benthic microbial populations was tested.[268] Toxicities of 20 dispersants to Hong Kong harbor fish were reported.[269] Tests with dispersant and surface-active-agent toxicity for 18 marine species showed that toxicity was more dependent on the species tested, rather than the product.[270] Two scallop predator species, *Asterias* and *Urosalpinx*, exhibited marked differences in predatory behavior to crude oil or crude oil/dispersant mixtures.[271]

Microbes. Metabolic studies with marine microbes and aliphatic hydrocarbons included evidence for anaerobic C_2 generation by sediment-dwelling methanogenic bacteria,[272] fatty acid production by *Nocardia* sp. when grown on kerosene,[273] and the presence of neosynthesized long-chain alkanes up to NC-56 in estuarine sediments following the *Amoco Cadiz* spill.[274] Biodegration of beached *Amoco Cadiz* oil led to an enrichment of isoprenoid alkanes, naphthenoaromatic and naphthenic compounds, alkylated phenanthrenes, aromatic organic sulfur compounds, hopanes, and C_{27}-C_{31} alkanes.[275] Sediment microbial degradation rates for benzo(a)pyrene were highest at pH 8.0.[276]

Table 5—Chemical residues reported from marine organisms.

Element	Type of organism	Concentration (μg/g dry wt); locality
Aluminum	Pelecypods	38.2–441.4; Irel.[179]
	Birds	0; Hawaii[179]
Arsenic	Phytoplankton	20.03–1 270[b]; Ga.[180]
	Brown algae	8.1–15.9; Austl.[181]
	Brown algae	11–160; U. K.[182]
	Marine grasses	0.1–0.59; Austl.[181]
	Polychaetes	2.4–31.3; Austl.[181]
	Polychaetes	4–87; U. K.[182]
	Pelecypod	5–190; U. K.[182]
	Gastropod	2.3; Austl.[181]
	Holothurians	4.1; Austl.[181]
	Fish	<0.025–14.5; Austl.[181]
Barium	Birds	0; Hawaii[179]
Bromine	Red algae	324–1 137[b]; Mass.[183]
	Mammals	5.5–42.8[a]; Neth.[184]
	Mammals	4.6–28.6[a]; Den.[184]
Cadmium	Macrophytes	210–500; Baltic Sea[185]
	Green algae	0.22–0.6; Tex.[222]
	Green algae	0.12–0.22[b]; Ga.[223]
	Spermatophyta	0.2–3.5; Tex.[222]
	Spermatophyta	0.12–23[b]; Mass.[186]
	Spermatophyta	0.12–1.3; U. S. Atlantic, P. R.[187]
	Polychaetes	0.26–0.48[b] New Brunswick[188]
	Polychaetes	0.9–40; U. K.[189]
	Pelecypods	0.19–0.29; New Brunswick[188]
	Pelecypods	0.2–2.3; Irel.[178]
	Pelecypods	130–560; Baltic Sea[185]
	Pelecypods	0.2–17.3; U. K.[190]
	Pelecypods	0.19–0.48; Isr.[191]
	Pelecypods	1.6–5.6; Hong Kong[192]
	Pelecypods	9.8–53; Austl.[193]
	Pelecypods	2.77–8.21; Thailand[194]
	Pelecypods	1.12–3.2; Canadian Atlantic[195]
	Pelecypods	0.09–10.78[a,b]; Mex.[196]
	Pelecypods	3.1–9.7[b]; Mass.[186]
	Pelecypods	0.4–1.5[b]; Mass.[183]
	Pelecypods	0–2.82; Mass.[197]
	Pelecypods	0.2–5.9[a,b]; Ala.[198]
	Pelecypods	0.44–1.5[a]; Calif.[199]
	Pelecypods	2.8–5; French Antilles[200]
	Gastropods	0.03–0.07[a]; Calif.[199]
	Gastropods	0–3.1; Irel.[178]
	Cephalopods	0.02–50; Monaco[201]
	Decapods	2.4–5.1; Irel.[178]
	Decapods	0.77–6.38[a,b]; French Mediterranean[202]
	Decapods	10.7–13[b]; Mass.[186]
	Decapods	0.003–1 600; Eastern Canada[203]
	Decapods	0.09–0.5[b]; Mass.[183]
	Decapods	0.17–0.77[b]; New Brunswick[188]
	Decapods	<0.01–0.04[a]; Calif.[199]
	Echinoidea	0.04–0.45[a]; Calif.[199]
	Fish	<0.01–0.07[a]; Calif.[199]
	Fish	3.4–4.4[b]; Mass.[186]
	Fish	0.02–0.1[b]; Mass.[183]
	Fish	1.7–122; Atlantic Ocean[204]
	Fish	0.3; French Antilles[200]
Cadmium	Fish	2–200[a]; Baltic Sea[185]
	Birds	0.000 2; Hawaii[179]
Cesium	Zooplankton	2.2–22.4[a]; Calif.[205]
	Zooplankton	2.77–6.15[a]; Tropical Pacific[205]
	Cephalopods	<2.3–5.1[a]; Calif.[205]
	Cephalopods	2.35–6.4[a]; Tropical Pacific[205]
	Fish	5–129[a]; Calif.[205]
	Fish	16.9–75.8; Tropical Pacific[205]
	Mammals	16–44.6[a]; Calif.[205]
Chromium	Brown algae	<0.5; Calif.[187]
	Spermatophyta	<0.05–10; U. S. Atlantic, Pacific, P. R.[187]
	Spermatophyta	1–33; Oreg.[206]
	Spermatophyta	2.3–44[b]; Mass.[186]
	Pelecypods	<0.01–0.77[a]; Calif.[199]
	Pelecypods	0.85–1.29[b]; Mass.[186]
	Pelecypods	0.5–10.3; Irel.[178]
	Pelecypods	0.11–0.3; French Antilles[200]
	Pelecypods	35–188; Hong Kong[192]
	Gastropods	<0.04–2.2[a]; Calif.[199]
	Gastropods	0–16; Irel.[178]
	Decapods	0–0.7; Irel.[178]
	Decapods	5.7–6.7[b]; Mass.[186]
	Decapods	<0.02–0.08[a]; Calif.[199]
	Echinoidea	0.04–0.19[a]; Calif.[199]
	Fish	<0.01–0.12[a]; Calif.[199]
	Fish	4.8–6.4[b]; Mass.[186]
	Fish	0.27–0.5; French Antilles[200]
	Birds	0; Hawaii[179]
Cobalt	Brown algae	1.8; Calif.[187]
	Red algae	1.5–4.6[b]; Mass.[183]
	Spermatophyta	6.3; P. R.[187]
	Pelecypods	0.2–19.9; U. K.[190]
Copper	Macrophytes	0.1–5[a]; Baltic Sea[185]
	Green algae	7.5–12.5; Tex.[222]
	Brown algae	5; Calif.[187]
	Red algae	1.5–223[b]; Mass.[186]
	Marsh grass	1.7–15; N. C.[207]
	Spermatophyta	4.5–20; N. C.[208]
	Spermatophyta	3–13[b]; Mass.[186]
	Spermatophyta	3–13; U. S. Atlantic, Pacific, P. R.[187]
	Spermatophyta	5–20; Tex.[222]
	Spermatophyta	5–26; Oreg.[206]
	Polychaetes	12–3 124; U. K.[209]
	Polychaetes	9.6–16.4[b]; New Brunswick[188]
	Polychaetes	5–47; U. K.[189]
	Pelecypods	4.8–21; Austl.[193]
	Pelecypods	3.75–16.9; Thailand[194]
	Pelecypods	192–2 296; Hong Kong[192]
	Pelecypods	0.08–0.63[a]; Calif.[199]
	Pelecypods	11–15.8[b]; Mass.[186]
	Pelecypods	9.7–517[a]; Mass.[183]
	Pelecypods	6.5–33.5; Mass.[197]
	Pelecypods	12.1–18.3[b]; New Brunswick[188]
	Pelecypods	14–500; Can.[210]
	Pelecypods	1–16[a]; Baltic Sea[185]

Table 5—(Continued)

Element	Type of organism	Concentration (μg/g dry wt); locality
Copper	Pelecypods	0.8–36.2; U. K.[190]
	Pelecypods	6.13–35.37; Isr.[191]
	Pelecypods	8–220.9; Irel.[178]
	Pelecypods	11.2–28.4; French Antilles[200]
	Gastropods	1.5–5.9[a]; Calif.[199]
	Gastropods	17.7–88.2; Irel.[178]
	Cephalopods	26–25 000; Monaco[201]
	Decapods	33.4–133.9; Irel.[178]
	Decapods	58–100.8[b]; New Brunswick[188]
	Decapods	136–155[b]; Mass.[183]
	Decapods	162–241[b]; Mass.[186]
	Decapods	2.5–22[a]; Calif.[199]
	Echinoidea	0.19–0.38[a]; Calif.[199]
	Fish	<0.02–4.3[a]; Calif.[199]
	Fish	26.6–30.1[b]; Mass.[186]
	Fish	0.4–1.7[b]; Mass.[183]
	Fish	0.08–2.4[a]; Baltic Sea[185]
Iodine	Red algae	58–371[b]; Mass.[186]
	Pelecypods	2.3–6.4[b]; Mass.[186]
	Decapods	6.3–8.9[b]; Mass.[186]
	Fish	0.6–0.9[b]; Mass.[186]
Iron	Brown algae	76; U. S. Atlantic, Pacific, P. R.[187]
	Green algae	160–250[b]; Ga.[223]
	Green algae	102–1 200; Tex.[222]
	Marsh grass	190–21 000; N. C.[207]
	Spermatophyta	590–6 000; N. C.[208]
	Spermatophyta	65–2 500[b]; Mass.[186]
	Spermatophyta	60–3 100; U. S. Atlantic, Pacific P. R.[187]
	Spermatophyta	100–12 800; Oreg.[206]
	Spermatophyta	90–1 600; Tex.[222]
	Polychaetes	239–1 730; U. K.[209]
	Pelecypods	185–217[b]; Mass.[186]
	Pelecypods	159–1 740[b]; Mass.[186]
	Pelecypods	1 240–23 100; Mass.[187]
	Pelecypods	343–4 165; Irel.[178]
	Pelecypods	43.7–120.8[b]; South Africa[211]
	Pelecypods	707–2 072; Hong Kong[192]
	Gastropods	682–3 107; Irel.[178]
	Cephalopods	10–700; Monaco[201]
	Decapods	229–1 345; Irel.[178]
	Decapods	50–202[b]; Mass.[183]
	Decapods	273–303[b]; Mass.[186]
	Fish	1 070–1 080[b]; Mass.[186]
	Fish	83–127[b]; Mass.[183]
	Birds	0.35; Hawaii[179]
Lead	Macrophytes	<0.002–5[a]; Baltic Sea[185]
	Brown algae	0.8; Calif.[187]
	Spermatophyta	2.2–97; U. S. Atlantic, Pacific[187]
	Spermatophyta	20–26[b]; Mass.[186]
	Polychaetes	0.42–2.06[b]; New Brunswick[188]
	Polychaetes	8–39; U. K.[190]
	Pelecypods	1.2–9.7; Austl.[193]
	Pelecypods	5.8–24.1; Hong Kong[192]
	Pelecypods	7.8–29.9; French Antilles[200]
	Pelecypods	<0.03[a]; Calif.[199]
	Pelecypods	6.2–10.1[b]; Mass.[186]
Lead	Pelecypods	0.31–2[b]; Mass.[183]
	Pelecypods	0–8.56; Mass.[197]
	Pelecypods	2.07–3.94[b]; New Brunswick[188]
	Pelecypods	9.9–20.2; Irel.[178]
	Pelecypods	0.7–59.4; U. K.[190]
	Pelecypods	0.16–1.56[a]; Baltic Sea[185]
	Pelecypods	1.56–13.01; Isr.[191]
	Gastropods	<0.08[a]; Calif.[199]
	Gastropods	3.4–22.4; Irel.[178]
	Decapods	37.6–43.9; Irel.[178]
	Decapods	0.04–0.47[b]; New Brunswick[188]
	Decapods	0.2–1.5[b]; Mass.[183]
	Decapods	116–121[b]; Mass.[186]
	Decapods	<0.14[a]; Calif.[199]
	Echinoidea	<0.01[a]; Calif.[199]
	Fish	<0.08[a]; Calif.[199]
	Fish	2.9–23.9; French Antilles[200]
	Fish	43.8–45.1[b]; Mass.[186]
	Fish	0.5–2.3[b]; Mass.[183]
	Fish	0.01–1.4[a]; Baltic Sea[185]
	Birds	0; Hawaii[179]
Magnesium	Birds	19.04; Hawaii[179]
Manganese	Green algae	20–45; Tex.[222]
	Green algae	13–22[b]; Ga.[223]
	Brown algae	5.5; Calif.[187]
	Marsh grass	11–280; N. C.[208]
	Spermatophyta	6–360; N. C.[207]
	Spermatophyta	3–50[b]; Mass.[186]
	Spermatophyta	22–561; U. S. Atlantic, Pacific, P. R.[187]
	Spermatophyta	30–593; Oreg.[206]
	Spermatophyta	5–280; Tex.[222]
	Polychaetes	5–20.7; U. K.[209]
	Pelecypods	1–206; U. K.[190]
	Pelecypods	11–131.1; Irel.[178]
	Pelecypods	5.8–6.3[b]; Mass.[186]
	Pelecypods	7.3–73[b]; Mass.[183]
	Pelecypods	24.5–63.7; Hong Kong[192]
	Gastropods	33.4–131.9; Irel.[178]
	Cephalopods	30–20; Monaco[201]
	Decapods	58.3–673.3; Irel.[178]
	Decapods	15.8–22.3[b]; Mass.[186]
	Decapods	7.1–12[b]; Mass.[183]
	Fish	3.9–5.2[b]; Mass.[183]
	Fish	22.4–25.1[b]; Mass.[186]
	Birds	0.003 6; Hawaii[179]
Mercury	Macrophytes	0.001–0.18[a]; Baltic Sea[185]
	Green algae	0–0.28; Isr.[212]
	Brown algae	0–0.065; Isr.[212]
	Brown algae	<0.4; Calif.[187]
	Marsh grasses	0.04–25; New England[213]
	Spermatophyta	0.12; P. R.[187]
	Zooplankton	0.49–0.95; India[214]
	Benthic Inverts.	2.61–3.03; India[214]
	Pelecypods	0.01–0.29; Thailand[194]
	Pelecypods	0.01–0.07[a]; Calif.[199]
	Pelecypods	305–568; New England[213]

(Table 5 Continued)

Table 5—(Continued)

Element	Type of organism	Concentration (μg/g dry wt); locality
Mercury	Pelecypods	0.004–0.77[a]; U. K.[190]
	Pelecypods	0.004–0.3[a]; Baltic Sea[185]
	Pelecypods	0.04–0.91[a]; Israel[191]
	Pelecypods	0.33[a]; Red Sea[215]
	Gastropods	0–0.735; Isr.[191]
	Gastropods	<0.01–0.06[a]; Calif.[199]
	Decapods	0.02–0.48[a]; Calif.[199]
	Decapods	0.25[a]; Red Sea[215]
	Decapods	57; New England[213]
	Echinoidea	<0.01–0.03[a]; Calif.[199]
	Cartilaginous fish	0.03–151; Austl.[216]
	Teleost fish	0–3.53; Austl.[216]
	Fish	0.1–1.5: Austl.[217]
	Fish	0.04–0.79[a]; Calif.[199]
	Fish	0.004–0.88[a]; Baltic Sea[185]
	Fish	0.55–5.67[a]; So. Africa[218]
	Fish	0.5[a]; Red Sea[215]
	Birds	0.5–2; Peru[219]
	Birds	0.000 9; Hawaii[179]
	Mammals	0.21–16.2; Eastern Canada[220]
	Mammals	0.7–573[a]; Netherlands[184]
	Mammals	0.5–751[a]; Denmark[184]
Molybdenum	Birds	0; Hawaii[179]
Nickel	Green algae	5.3–6.7; Texas[222]
	Spermatophyta	1–4.5; Texas[222]
	Spermatophyta	2.6–6.9; U. S. Atlantic, Pacific, P. R[187]
	Pelecypods	0.02–0.47[a]; Calif.[199]
	Pelecypods	1.7–6.5[b]; Mass.[183]
	Pelecypods	0.3–35.4; U. K.[190]
	Pelecypods	0.9–6.1; Irel.[178]
	Gastropods	0.4–15.6; Irel.[178]
	Gastropods	0.14–2[a]; Calif.[199]
	Decapods	<0.04–0.51[a]; Calif.[199]
	Decapods	1.4–5.7[b]; Mass.[183]
	Decapods	9.6–15.6; Irel.[178]
	Echiniodea	<0.02–0.13[a]; Calif.[199]
	Fish	<0.01–0.67[a]; Calif.[199]
	Fish	0.5[b]; Mass.[183]
	Birds	0; Hawaii[179]
Potassium	Zooplankton	1.48–1.73[a]; Calif.[205]
	Zooplankton	0.8–1.98[a]; Tropical Pacific[205]
	Cephalopods	1.1–2.6[a]; Calif.[205]
	Cephalopods	2.5–3.73[a]; Tropical Pacific[205]
Rubidium	Green algae	12–18; Ga.[223]
	Fish	2.77–5.8[a]; Calif.[205]
	Fish	2.54–5.33[a]; Tropical Pacific[205]
	Mammals	2.2–4.3[a]; Calif.[205]
Selenium	Pelecypods	0.9–1.6[b]; Mass.[183]
	Decapods	2.2–3.1[b]; Mass.[183]
	Fish	0.8–1.3[b]; Mass.[183]
	Mammals	Tr-350[a]; Neth.[184]
	Mammals	Tr-409[a]; Den.[184]
Silver	Polychaetes	<0.5–5; U. K.[209]
	Pelecypods	1–23; Can.[210]
	Gastropods	<0.01–0.06[a]; Calif.[199]
	Decapods	<0.02–0.2[a]; Calif.[199]
	Echinoidea	<0.02–0.01[a]; Calif.[199]
	Fish	<0.01–0.06; Calif.[199]
Strontium	Brown algae	330; Calif.[187]
	Spermatophyta	24–369; U. S. Atlantic, Pacific, P. R.[187]
	Spermatophyta	18–260; Oreg.[206]
	Birds	0–25; Hawaii[179]
Vanadium	Pelecypods	20–139.2; French Antilles[200]
	Cephalopods	0.2–25; Monaco[201]
	Fish	14.4–171.5; French Antilles[200]
Zinc	Macrophytes	3–65; Baltic Sea[185]
	Green algae	40–80; Tex.[222]
	Green algae	6.2–12[b]; Ga.[223]
	Brown algae	36; Calif.[187]
	Red algae	19–188[b]; Mass.[183]
	Marsh grass	10–62; N. C.[208]
	Spermotophyta	37–240; N. C.[207]
	Spermatophyta	5–90; Tex.[222]
	Spermatophyta	4.5–60; U. S. Atlantic, Pacific, P. R.[187]
	Spermatophyta	26–62; Oreg.[206]
	Spermatophyta	30–150[b]; Mass.[186]
	Polychaetes	63–258; U. K.[209]
	Polychaetes	41–400; U. K.[188]
	Polychaetes	131.6–245.6[b]; New Brunswick[188]
	Pelecypods	105–410; Austl.[193]
	Pelecypods	47.6–91.7; Thailand[194]
	Pelecypods	20–33[a]; Calif.[199]
	Pelecypods	61.7–62.6[b]; Mass.[186]
	Pelecypods	115–3 856[b]; Mass.[183]
	Pelecypods	1.89–309; Mass.[197]
	Pelecypods	178.6–218.8[b]; New Brunswick[188]
	Pelecypods	733–1 360; Hong Kong[192]
	Pelecypods	47.6–474.5; Irel.[178]
	Pelecypods	16–110; Baltic Sea[185]
	Pelecypods	16.1–634; U. K.[190]
	Pelecypods	51.2–141.6; Isr.[191]
	Cephalopods	50–1 450; Monaco[201]
	Gastropods	65.3–334.3; Irel.[178]
	Gastropods	4.3–7.1[a]; Calif.[199]
	Decapods	6–210[a]; Calif.[199]
	Decapods	146–150[b]; Mass.[186]
	Decapods	135–207[b]; Mass.[183]
	Decapods	50.4–81.3[b]; New Brunswick[188]
	Decapods	75.6–142; Irel.[178]
	Echinoidea	2.8–41[a]; Calif.[199]
	Fish	<1.2–6.6[a]; Calif.[199]
	Fish	222–241[b]; Mass.[186]
	Fish	73–109[b]; Mass.[183]
	Fish	1.2–32; Baltic Sea[185]
	Fish	61–3 300[b]; Sp.[221]
	Birds	0.85–Hawaii[179]

[a] Wet weight.
[b] Experimental study.

Bioaccumulation. An overview on the uptake, disposition, and effects of petroleum hydrocarbons in marine fish was presented.[277] Uptake and intrahistological transport of naphthalene from seawater was examined in Pacific oyster, *Ostrea edulis*.[278,279] Depuration of No. 2 fuel oil hydrocarbons from stony coral was incomplete for up to 2 weeks, following experimental exposure in Caribbean waters.[280] Results were given of a study on the uptake of benzo(a)pyrene by a diatom and the transfer of this hydrocarbon to a larval bivalve fed a diet consisting solely of the diatom.[281]

Characteristics of mixed function oxygenases among marine invertebrates were reviewed by Lee.[282] Sponges (*Tethya lyncurium*) apparently lack a mixed-function oxidase system.[283] This hydrocarbon-degrading system was detailed for several species of crabs, a polychaete[284] and shrimp.[285] The highly efficient detoxication systems of starry flounder and English sole were detailed in separate studies.[286,287]

Field studies. Simulated oil spills were studied that examined the risks to neighboring coastal areas,[288] the responses of salt marsh ecosystems,[289] and repeated dosing with a No. 2 fuel oil.[290] The macrofauna and meiofauna of three oiled and three control experimental ecosystems were followed for 25 weeks of semicontinuous additions of an oil-water dispersion of No. 2 fuel oil.[291]

Reports of effects resulting from nonexperimental chronic oil pollution included an overview of surface oil pollution in the Indian Ocean.[292] Results of histopathological surveys for oil pollution effects in north and southeastern U. S. waters were given.[293,294] An examination of the physical effect of oil in reducing the filtration or demineralization processes of interstitial pore systems was made.[295]

Biological perspectives of catastrophic spillage were presented.[296,297] Continued analyses of the effects oil spills included the *Amoco Cadiz*,[298-304] *Eleni V*,[305,306] *Bohlen*,[307] *Ixtoc I*,[308] and spills in the Baltic Sea[309] and Texas.[310] Effects of oil on meiofauna of South African beaches revealed that removal of oil-sand debris had a greater effect on the density of copepods and nematodes than did oil.[311]

Toxicity and sublethal effects. Toxicity of a simulated refinery effluent and its components on mysid shrimps was more sensitive than to estuarine fish and grass shrimp as well as 17 freshwater organisms.[312] Lee and Nicol[313] showed that two ecologically related amphipods exhibited marked differences in embryo-sensitivity when tested with water-soluble fuel oil. Toxicity of artificially weathered oil to horseshoe crab juveniles was studied.[314] Little relationship between the accumulation of di- and triaromatic compounds in shrimp tissues, and subsequent cumulative mortality during long-term exposure, was noted by Anderson *et al.*[315]

The applicability of measuring food assimilation and respiration efficiencies as measures of severity of oil pollution was discussed.[316] Effects of petroleum hydrocarbons on natural assemblages of mixed phytoplankton[317] and zooplankton[318] were studied. The construction, operation, and performance of a continuous flow oil dosing system was detailed.[319]

Reports of sublethal effects of petroleum hydrocarbons concerned numerous plant and animal groups including protozoa,[320] diatoms,[321-323] brown algae,[324] and polychaete.[325] Physiological parameters among mollusks that were affected by petroleum hydrocarbons included glucose metabolism,[326] lysosomal structure,[327] respiratory rate,[328] and amino acid metabolism.[329] Reports of hydrocarbon influences on crustaceans included ghost crab reproductive behavior,[330] respiration and growth in mud crabs,[331] thermo- and halo-tolerance in grass shrimp,[332] haemocyanin-O_2 binding capacity in rock crab,[333] feeding among zooplankters,[334] nesting behavior and storage lipid deposition in an amphipod,[335] and feeding behavior in an amphipod.[336] Data were presented for effects of drilling fluid,[337] sunken crude oil,[338] and diesel fuel[339] on echinoderm species. Selected histochemical parameters of liver sections from control and WSF-exposed fish were evaluated,[340] as was the modification of predatory behavior of coho salmon by crude oil.[341] The effects of petroleum hydrocarbons on the growth and/or reproduction of the following fish species were reported: killifish (*Fundulus*),[342] flagfish (*Jordanella*),[343] red drum (*Sciaenops*),[344] winter flounder (*Pseudopleuronectes*),[345] and coho salmon (*Oncorhynchus*).[346]

Aves. Ingestion of small, sublethal amounts of bunker C and Prudhoe Bay crude oil reduced the proportion of Cassin's auklet laying eggs, but chick development after hatching was unaffected by any treatment.[347] Dosing with oil/dispersant emulsion had no greater effect on weight loss among nestling herring gulls than oil alone.[348] Pathological and reproductive biochemical alterations were no greater in mallards treated for 26 weeks with a diet of 0.25 or 2.5% crude oil than in controls.[349] Rattner[350] reported that adult mallards may be able to tolerate large quantities of crude petroleum mixed in their diet without overt or biochemical indications of distress. Liver hypertrophy and splenic atrophy were gross evidences of pathological effects in ducklings fed 5% fuel oil diets for 18 weeks.[351] Decreased ovarian activity in oil-fed Khaki Campbell ducks was not due to antigonadal effects of increased prolactin secretion or adrenocortical activity; likewise, an endocrine dysfunction of the thyroid was concluded unlikely to be casually responsible for the effects of petroleum on avian reproduction.[352] Gas chromatographic mass spectral analysis of externally treated mallard eggs showed the passage of aromatic hydrocarbons through the shell and shell membranes to developing embryos.[353]

COMPLEX EFFLUENTS AND ENVIRONMENTS

Klumpy and Peterson[354] found that the algae *Fucus spiralis* assimilated arsenic mainly as 1 lipid-soluble compound and 12 minor water-soluble organo-arsenic compounds. The snail *Littorina littoralis* that eats *Fucus* produces one major organo-arsenic compound. The water soluble organo-arsenic compounds found in this study were different from those previously found or proposed in marine organisms. Edmonds and Francesconi[355] isolated two main arsenical constituents of the brown kelp *Ecklonia radiata*. These compounds were 2-hydroxy-3-sulphopropyl-5-deoxy-5-(dimethyl-arsenoso) furanoside and 2,3-dihydroxypropyl-5-deoxy-5-(dimethyl-arsenoso) furanoside. They believed that these compounds could be further metabolized to arsenobetaine and are therefore intermediates in arsenate (in water) to arsenobetaine (in organisms).

It was found that boron may alleviate nutrient deficiency and cause temporal variations of phytoplankton composition in coastal waters.[356] Sanders *et al.*[357] showed that chlorine

Table 6—Effects of trace contaminants on marine organisms.

Toxicant	Group	Species	Observed effects	Effective concentration (mg/l)	Reference number
Ar	Oysters	*Crassostrea gigas* (embryos)	48h EC50	0.326	379
	Mussels	*Mytilus edulis* (embryos)	48h EC50	>3	
	Crabs	*Cancer magister* (larvae)	96h LC50	0.232	
B	Phytoplankton		168% stimulation of primary production (Nov.–May)	30	356
			Inhibited photosynthesis by 30% (June–Oct.)	30	
Ca	Oysters	*Crassostrea gigas* (embryos)	48h EC50	0.611	379
	Mussels	*Mytilus edulis* (embryos)	48h EC50	1.2	
	Crabs	*Cancer magister* (larvae)	96h LC50	0.247	
Ca	Crabs	*Cancer irroratus*	37% inhibition of Na^+-K^+-ATPase	10	380
			Inhibition of Mg^{+2} −ATPase	1	
Cl	Fish	*Chromis punctipinnis*	Active avoidance of chlorinated seawater	0.15–0.16	381
Cr VI	Polychaetes	*Neanthes arenaceodentata*	Cessation of reproduction	0.1	382
			Reduction of numbers of offspring produced	0.012–0.05	
Cr VI	Oysters	*Crassostrea gigas* (embryos)	48h EC50	4.538	379
	Mussels	*Mytilus edulis*	48h EC50	4.469	
	Crabs	*Cancer magister*	96h LC50	3.44	
Cr III	Polychaetes	*Neanthes arenaceodentata*	No adverse effects observable	50.4	382
Chrome ligno-sulphonate drilling mud (mud aqueous fraction, MAF)	Polychaetes	*Neanthes arenaceodentata*	96h LC50	10–51% MAF	360
		Ophryotrocha labronica		>100% MAF	
		Dinophilus sp.		76% MAF	
		Ctenodrilus serratus		13–32% MAF	
	Mysids	*Mysidopsis almyra* (1-day-old)	96h LC50	27% MAF	
	Shrimp	*Palaemonetes pugio* (4-day zoeae)	96h LC50	34% MAF	
		Penaeus duoarum (post-larvae)		86% MAF	
Cu	Oysters	*Crassostrea gigas* (embryos)	48h EC50	0.005 3	379
	Mussels	*Mytilus edulis* (embryos)	48h EC50	0.005 8	
	Crabs	*Cancer magister* (larvae)	96h LC50	0.049	
	Clams	*Meretrix casta*	96h LC50	0.57	383
	Amphipods	*Gammarus duebeni*	96h LC50	>10	367

Table 6—(Continued)

Toxicant	Group	Species	Observed effects	Effective concentration (mg/l)	Reference number
Cu	Clam	*Scrobicularia plana* (isolated siphon tissue)	Caused inhalant siphon to contract	0.25–0.5	384
	Mussel	*Mytilus galloprovincialis*	Induction of the synthesis of copper-binding proteins similar to metallothioneins	0.08	385
Fl	Brine shrimp	*Artemia salina*	Reduction in 12-day growth	5	386
	Phytoplankton	*Dunaliella tertiolecta*	Growth inhibition	0.2	387
		Pavlova lutheri	35–50% growth inhibition	0.15–0.2	
		Amphidinium carteri	20–25% growth inhibition	0.15	
		Thalassiosira weissflogii	11% growth inhibition	0.2	
		Chaetoceros gracilis	26% growth stimulation	0.2	
Pb	Oysters	*Crassostrea gigas* (embryos)	48h EC50	0.758	379
	Mussels	*Mytilus edulis* (embryos)	48h EC50	0.476	
	Crabs	*Cancer magister*	96h LC50	0.575	
Pb	Crabs	*Cancer irroratus*	Inhibition of Mg^{+2}-ATPase	1	380
Hg	Periphyton		Changed growth pattern reduced number of species	0.005	388
			Inhibition of growth	0.05	
Hg	Oysters	*Crassostrea gigas* (embryos)	48h EC50	0.006 7	379
	Mussels	*Mytilus edulis* (embryos)	48h EC50	0.005 8	
	Crabs	*Cancer magister* (larvae)	96h LC50	0.008 2	
Hg	Amphipods	*Gammarus duebeni*	96h LC50	0.122	367
			96h LC50 (after pretreatment with 1.8 mg Cu/l	0.165	
Ni	Oysters	*Crassostrea gigas* (embryos)	48h EC50	0.349	379
	Mussels	*Mytilus edulis* (embryos)	48h EC50	0.891	
	Crabs	*Cancer magister* (larvae)	96h LC50	4.36	
Ozone-produced oxidants	Fish	*Morone saxatilis*	96h LC50	0.08	
Ozone (OPO)	Crabs & Fish	*Callinectes sapidus*	96h LC50	0.26–0.28	389
		Brevoortia tyrannus	96h LC50	0.36	
Se	Oysters	*Crassostrea gigas* (embryos)	48h EC50	>10	379
	Mussels	*Mytilus edulis* (embryos)	48h EC50	>10	
	Crabs	*Cancer magister* (larvae)	96h LC50	>10	

(Table 6 Continued)

Table 6—(Continued)

Toxicant	Group	Species	Observed effects	Effective concentration (mg/l)	Reference number
Se	Mysids	*Mysidopsis bahia* (subadult)	96h LC50	1.5	390
	Shrimp	*Penaeus aztecus*	96h LC50	1.2	
	Crabs	*Callinectes sapidus* (juveniles)	96h LC50	4.6	
	Fish	*Cyprinodon variegatus* (juveniles)	96h LC50	7.4	
	Fish	*Lagodon rhomboides* (subadult)	96h LC50	4.4	
Ag	Oysters	*Crassostrea gigas* (embryos)	48h EC50	0.022	379
	Mussels	*Mytilus edulis* (embryos)	48h EC50	0.014	
	Crabs	*Cancer magister* (larvae)	96h LC50	0.055	
Sulfide in sediments	Amphipod	*Corophium volutator*	Avoidance of sulfide sediments	5.8×10^{-5} M	391
			Inhibition of burrowing	5.8×10^{-4} M	
Sulfide	Polychaetes	*Arenicola marina*	Survival time: 55–66 h	0.01 M	
		Nephtys circosa	Survival time: 48 h	0.01 M	
		Nephtys hombergi	Survival time: 72 h	0.01 M	
		Pectinaria koreni	Survival time: 59 h	0.01 M	
		Scolelepis squamata	Survival time: 65 h	0.01 M	
		Owenia fusiformis	Survival time: 128 h	0.01 M	
Zn	Oysters	*Crassostrea gigas* (embryos)	48h EC50	0.119	379
	Mussels	*Mytilus edulis* (embryos)	48h EC50	0.175	
	Crabs	*Cancer magister* (larvae)	96h LC50	0.456	

caused a rapid decline and copper a gradual decline in biomass and species diversity of phytoplankton assemblages, whereas a rise of 2°C in water temperature slightly increased biomass but did not change the diversity. Stress led to decreases in centric diatoms and predominance of microflaggellates. The accidental addition of 0.025 to 0.040 mg/l of copper to natural phytoplankton assemblages in experimental ecosystems led to blooms of *Amphiprora paludosa* var. *hyalina* that fluorished virtually as monocultures even after copper levels had decreased.[358] A study with phytoplankton *Navicula incerta* and *Tetraselmis maculata* and mussels *Mytilus edulis* showed that nitrosodimethylamine caused changes in the health of these organisms.[359]

Neff *et al.*[360] carried out bioassays with polychaetes, mollusks, and crustaceans to several fractions of chrome lignosulphonate drilling mud. They concluded that the discharge of this material from offshore platforms is not likely to cause measurable change to benthic, demersal, or pelagic marine animals.

Experiments with striped bass *Morone saxatilis* eggs and fingerlings showed that the eggs were significantly more sensitive to ozone-produced oxidants in freshwater than seawater, and that the toxicities of chlorine and ozone were similar for eggs in estuarine waters.[361] Ozone and chlorine are capable of altering the pH of seawater. The effects of pH change on marine organisms, however, are inadequately known.[362]

Roesijadi[363] reviewed the information available on the metal-binding protein metallothionein in marine invertebrates. He covered the properties, occurrence, and the implication of metallothionein's role in metal toxicity. Metallothionein has a high capability of forming sulfhydryl bonds with several metals and other compounds. The relative toxicities of mercury, copper, zinc, lead, cadmium, and manganese were shown to be correlated with solubility products of corresponding metal sulfides; and it was suggested that a common action for these metals had the toxicity deriving from metal-binding to sulfhydryl-containing compounds in the cell.[364] Experiments with copper and zinc and the diatom *Asterionella japonica* demonstrated that these metals caused the uncoupling of photosynthesis from cell division. These results suggested that toxic metals may bind to sulfhydryl groups on cell membranes, thereby impairing normal membrane function and reducing silicic acid uptake and amino acid synthesis, resulting in depressed cell division rates.[365] However, in another study, Rueter and Morel[366] showed that zinc-deficient *Thalassiosira pseudonana* had reduced silicic acid uptake rates. After addition of zinc to the medium to compensate for deficiency, copper toxicity was found to also reduce silicic acid uptake.

They found that the uptake rate was a function of the ratio in the medium of the cupric ion activity to zinc activity rather than either metal separately, and proposed an interaction site between silicic acid and cupric ion activity involving a zinc-dependent active site.

The effects of mercury and copper on urine production of the amphipod *Gammarus duebeni* were less with the two metals together than the additive separate effects. Pre-exposure to copper or simultaneous exposure to copper significantly reduced mercury accumulation.[367] Copper and mercury were far more toxic than zinc to the alga *Ascophyllum nodosum*, but zinc and copper or zinc and mercury were considerably less toxic than what additivity should have been. Zinc was antagonistic to copper and mercury.[368] Other evidence of metal-metal antagonism was shown with the archiannelid *Dinophilus gyrociliatus*. Cadmium and mercury had antagonistic toxic effects at a salinity of 38 ppt but were synergistic at 25 ppt.[369] Copper and zinc, and copper, zinc, and mercury mixtures had synergistic toxicity effects on 48-hour development of oysters.[370] A low molecular weight metal-binding protein, not characteristic of metallothionein, was isolated from the gills of the mussel *Mytilus edulis*;[371] and lead incorporated into sea urchin embryos has been found to be associated with three discrete protein peaks when the urchin cytosol was further fractionated by gel filtration chromatography.[372]

The mixed function oxygenase (MFO) system in marine invertebrates, which metabolizes both environmental chemicals and steroid hormones has been reported in 18 species of marine invertebrates belonging to 4 phyla (mollusca, annelida, echinodermata, and arthropoda). The marine invertebrate MFO system is a multicomponent system of phospholipid NADPH cytochrome, P-450 reductase, and cytochrome P-450.[373]

Two types of effluents have been tested. Bleached kraft mill effluent toxicity was found to be a function of the developmental stage of the sheepshead minnows.[374] Sea urchin eggs, sperm, embryos, and larvae were utilized in the measurement of municipal wastewater effluent toxicity and found to be sensitive indicators of this effluent.[375]

Experiments were conducted to determine whether diatom clones from waters chronically contaminated with metals were more metal resistant than clones of the same species isolated from cleaner waters. It was found that metal additions rich in dissolved organic compounds would likely have less effect on resident phytoplankton than similar additions to waters low in organics, regardless of whether the clones were from different sites.[376]

Copper concentration, oxygen consumption, flesh weight, and shell size and weight were determined for *Mytilus edulis* and *Littorina rudis* from sites at different distances from a copper-laden estuary. The flesh weight of *Mytilus* decreased as the estuary was approached.[377] Two populations of the copepod *Acartia clausi* were studied; it was found that pollution-adapted specimens had higher ingestion and feeding rates, longer adult life span, and increased fecundity.[378] Papers concerned with the toxicity of contaminants to marine and estuarine organisms are summarized in Table 6.

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